LCA Methodology

On the Meaning of the Distance-to-Target Weighting Method and Normalisation in Life Cycle Impact Assessment

Jyri Seppälä[1] and Raimo P. Hämäläinen[2]

[1]Finnish Environment Institute, PB 140, 00251 Helsinki, Finland
[2]Systems Analysis Laboratory, Helsinki University of Technology, Otakaari 1 M, 02150 Espoo, Finland

Corresponding author: Jyri Seppälä; e-mail: jyri.seppala@vyh.fi

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Abstract. Distance-to-target weighting methods are widely used in life cycle impact assessment. The methods rank impacts as being more important the further away society's activities are from achieving the desired targets for the pollutants. However, we feel that the scientific bases of the distance-to-target methods still need more clarification. This article illustrates how multiattribute value theory (MAVT) can be applied to interpret the impact category weights as well as the aggregation rule and normalisation used in the distant-to-target methods. Our comparison revealed that under certain conditions two of the three commonly used impact assessment methods (Ecoindicator 95, ET-method) applying distance-to-target weighting are consistent with the impact assessment framework derived from MAVT. This consistency holds for non-zero targets with equal importance and linear damage functions passing through the origin. We show that the MAVT framework offers a foundation for the methodological development in life cycle impact assessment.

Keywords: Comparisons; damage functions; environmental impacts; LCA; Life Cycle Assessment (LCA); LCIA; Life Cycle Impact Assessment (LCIA); multiattribute value measurement; normalisation; theories; tradeoffs; weighting

Introduction

Life cycle assessment (LCA) has been widely used in analysing and assessing environmental impacts of materials, products or services over their entire life cycle. Despite the popularity of LCA, many authors have expressed concern over certain issues in the current practice of the life cycle impact assessment (LCIA) methodology. One fundamental question is the selection of impact category weights in order to aggregate the total impact from various impact categories into a single score. There are three commonly used groups of weighting methods (Finnveden 1996): the panel method, the monetary method and the distance-to-target method. Weighting according to the distance-to-target has been used in many popular LCIA methods (e.g. Ahbe et al. 1990, Baumann and Rydberg 1994, Goedkoop 1995).

In the distance-to-target approach, weights are derived from the extent to which actual environmental performance deviates from some goal or standard. As described by Powell et al. (1997), the method ranks impacts as being more important the further away society is from achieving the desired standard for the pollutant. However, the arguments for setting the targets are seldom transparent (Lindeijer 1996). They vary between countries and may be politically rather than scientifically based. Hird (1994) showed that politically determined targets are often agreed upon in an arbitrary fashion. Furthermore, distance-to-target methods are based on the assumption that all targets are equally important (Lindfors et al. 1995). According to Finnveden (1996) the distance-to-target methods are not weighting methods at all. They can be considered as a form of normalisation. According to Lee (1999), the original weighting factor in the distance-to-target method is not sufficient, as there is a need to use a correction factor that indicates the relative significance of an impact category with respect to other impact categories within a given safeguard area. We believe that Finnveden's and Lee's observations are correct, but there is still a continuing need for research which seeks to clarify interrelationships between the distance-to-target weighting and aggregated category indicator results.

This paper addresses the distance-to-target weighting method from the perspective of the multiattribute value theory (MAVT). Based on the work presented by Keeney and Raiffa (1976) and Dyer and Sarin (1979), MAVT can be used to help decision makers in multicriteria decisions. It offers a sound axiomatic foundation for aggregation and weighting in life cycle impact assessment. Seppälä (1997, 1999) showed that the life cycle impact assessment framework proposed by the Society of Environmental Toxicology and Chemistry (Consoli et al. 1993, Barnthouse et al. 1997, Udo de Haes et al. 1999) and the International Organisation for Standardisation (2000) can be constructed following the MAVT. Furthermore, Miettinen and Hämäläinen (1997, 1999) described ways to integrate MAVT and LCA. However, we believe that the aggregation and weighting procedures following MAVT have not yet been sufficiently well elaborated in the field of LCA. The aim of this paper is to familiarise the readers with the decision analytic approach, which integrates the normalisation and weighting concepts into LCA.

1 Structure of the Distance-to-Target Method

Several weighting methods are called 'distance-to-target methods' although the methods differ from each other both in the calculation rules and in the way targets are chosen. In the Ecoscarcity method (Ahbe et al. 1990), neither classification nor characterisation is performed and the total environmental impact caused by a product system a is computed from

$$I(a) = \sum_{j=1}^{m} \frac{L_j^N}{L_j^T} \cdot \frac{1}{L_j^T} \cdot c \cdot L_j(a) \qquad (1)$$

where

$I(a)$ = total environmental impact result caused by product system a
L_j^N = actual level of environmental intervention (emission) j related to a given region
L_j^T = target level of environmental intervention (emission) j related to a given region
$L_j(a)$ = amount of environmental intervention (emission) j caused by product system a (obtained from the inventory)
c = constant

The target levels are derived from annual load targets as set by national environmental protection agencies, laws and regulations. The distance-to-target factor (L_j^N / L_j^T) is multiplied with inventory data on environmental interventions and divided by the target level of interventions.

The framework of the Dutch and Swedish Environmental Theme (ET) method (McKinsey & Company 1991, Baumann and Rydberg 1994) and of the Ecoindicator 95 (Goedkoop 1995) method includes classification and characterisation. After these stages it is possible to calculate an indicator result for a given impact category i. The indicator result caused by product system a is calculated by

$$I_i(a) = \sum_{j=1}^{m} L_j(a) \cdot C_{i,j} \qquad (2)$$

where

$I_i(a)$ = indicator result of impact category i caused by product system a
$L_j(a)$ = amount of environmental intervention (emissions, extractions or land use) j caused by product system a
$C_{i,j}$ = characterisation (or equivalency) factor of intervention j within impact category i

Furthermore, an aggregating impact from various indicator result $I_i(a)$ into a single score $I(a)$ is calculated by the following equation:

$$I(a) = \sum_{i=1}^{n} w_i \frac{I_i(a)}{N_i} \qquad (3)$$

where

$I(a)$ = total environmental impact result caused by product system a
w_i = weight of impact category i
N_i = normalisation reference for impact category i

A normalisation reference is the total characterised impact indicator result calculated on the basis of an inventory of all the society's activities in some given area and over a reference period of time (Consoli et al. 1993, Wenzel et al. 1997).

According to the ET method the impact category weight w_i is defined as

$$w_i = \frac{N_i}{T_i} \qquad (4)$$

where T_i is a target reference of an impact category i. T_i can be defined as the level of impact indicator result at which no harmful effects are observed in a given environment. In this case determination of the value is based on ecological critical loads (sustainable loads). In practice, this is usually difficult due to incomplete information. In many cases, T_i's have been defined as politically maximum acceptable limits (political targets) (e.g. Baumann and Rydberg 1994).

In the Ecoindicator 95 method, the setting of sustainability targets involves acceptance of a certain level of ecological damage and impact category weight w_i is defined as

$$w_i = W_i \frac{N_i}{T_i} \qquad (5)$$

where W_i is a damage weighting factor. According to Goedkoop (1995), all damage weighting factors can be set to 1 when the target values for impact categories i are determined at the same damage level. The authors had considered three different damage levels to be equal (1 death per million inhabitants per year = impairment of 5% of the ecosystem = smog periods). Thus, after the determination of target values at the same damage level, the final weighting factors correspond to the so-called reduction factor (N_i / T_i).

2 Multiattribute Value Theory Framework

Multiattribute value theory (MAVT) refers to methods in multiattribute utility theory (MAUT) in which there are no uncertainties about the consequences of the alternatives (see, e.g. Keeney and Raiffa 1976, von Winterfeldt and Edwards 1986, French 1988, Keeney 1992). MAVT is based on the use of measurable value functions, which have special properties (see Krantz et al. 1971, Keeney and Raiffa 1976, Dyer and Sarin 1979). Measurability is needed when we need to express strengths of preference between attributes. In decision analysis, attributes are defined as measures for the achievement of the objectives. An objective is a statement of something that is desired to be achieved.

Most evaluation problems involve multiple attributes. It is assumed that the possible impact of selecting an alternative can be presented by the consequence $x = x_1,\ldots,x_n$ where x_i is defined to be a specific level of attribute i. In MAVT the starting point for obtaining the total impact of the alternative from the different consequences x_i are single-attribute value functions $v_i(x_i)$. By using the single-attribute value functions, the 'natural' scale of an attribute is converted into a value scale, which should reflect the decision maker's relative preferences for different levels of that attribute. The con-

sequences of each alternative are valuated by putting values x_i into the corresponding single-attribute functions $v_i(.)$. In addition, the assessment of the total impact value of an alternative requires the determination of the relative weights of the attributes. The weights can also be seen as scaling constants related to the different measurement scales of the attributes. Combining the value scores of the attribute consequences and weights according to the rules of MAVT produces total impact values $v(x)$ for the alternatives. These values reflect the decision maker's preferences between the alternatives when the axioms of rationality assumed by multiattribute value theory are accepted. An individual person or group of persons may have the role of a decision maker.

There are strong similarities between the MAVT approach to evaluate decision alternatives and life cycle impact assessment (LCIA). In general, the aim of LCIA is to provide an evaluation of the product system alternatives with regard to their potential environmental impacts. The starting point for aggregating the total impact from the different category indicator results I_i (i=1,..., n) into a single value score $I(.)$ can be described with help of damage functions $d_i(.)$. They describe the relationships between damages within impact categories i and category indicator results. A given working point of a product system a can be illustrated by using damage functions. The present level of a damage within impact category i caused by all the society's activities is obtained when a value of the normalisation reference (N_i) has been put into the damage function ($d_i(N_i)$). Furthermore, the contribution of the product system a to the damage is $d_i(N_i) - d_i(N_i - I_i(a))$, where $I_i(a)$ is a value of the category indicator result caused by the product system a (Fig. 1). Note that in LCIA it is usually assumed that the level of $I_i(a)$ is small compared with the level of N_i (see, e.g. Udo de Haes et al. 1999).

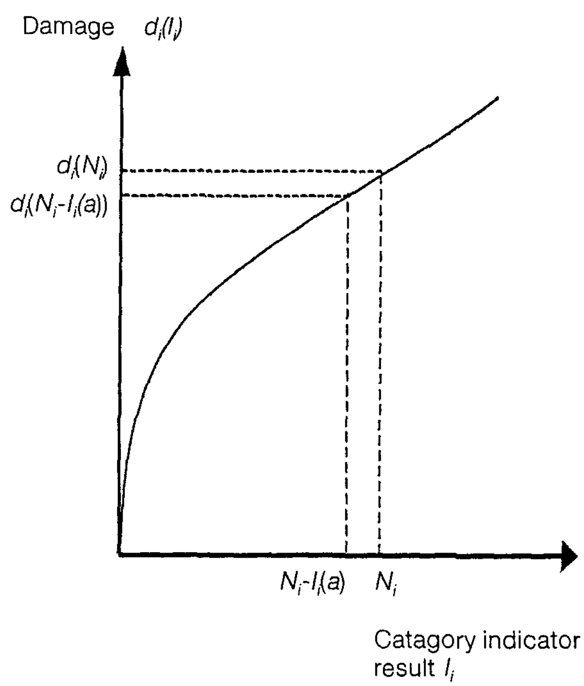

Fig. 1: Damage function (N_i = a normalisation reference for a given impact category i, $I_i(a)$ = a value of category indicator result I_i caused by the product system a)

A damage function is determined for each impact category as a single-attribute value function in the terminology of MAVT. In addition, a category indicator result I_i corresponds to the concept of an attribute. According to MAVT the calculation of the total environmental impact result $I(.)$ can be written as

$$I(I_1, ..., I_n) = f(d_1(I_1), ... d_n(I_n)) \tag{6}$$

The easiest form of Eq. 6 is the additive function

$$I(I_1, ..., I_n) = w_1 d_1(I_1) + ... + w_n d_n(I_n) \tag{7}$$

where w_is (i =1,...,n) are the weights of the single-damage functions (= single-attribute functions in the terminology of MAVT).

A sufficient condition for an additive decomposition of the multiattribute value function is mutual preferential independence of the attributes (Keeney and Raiffa 1976). This requires that preferences of disjoint subsets of the attributes are independent. For example, preferential independence between two attributes, category indicator results I_1 (e.g. aquatic eutrophication potential in PO_4 equivalents) and I_2 (e.g. acidification potential in SO_2 equivalents), would hold if the preferences for the specific value of attribute I_1 are not dependent on the value level of attribute I_2. If the value of attribute I_2 is also preferentially independent of the value of attribute I_1, then they are mutually, preferentially independent.

The single-attribute functions are normally normalised onto the [0,1] range. Once a suitable range of achievement levels [I_i^o, I_i^*] has been defined for each attribute, it is customary to normalise the value function so that the values $d_i(I_i^o)$=0 and $d_i(I_i^*)$=1 are assigned to the best and worst possible consequences.

Assuming that the conditions for an additive multiattribute function are met, the weights w_i can be assessed by a number of alternative decision analysis techniques available in the literature. In this paper, the so-called tradeoff procedure is used because of its strong theoretical foundation (Keeney and Raiffa 1976, Weber and Borcherding 1993). The ratio of two weights w_1 and w_2 can be obtained in the following way. First choose two points from the scale of attribute 1 and one from the scale of attribute 2 and denote them by I_1^1, I_1^2, I_2^1. The idea is to identify the second point y_2 on the scale of attribute 2 so that it makes the attribute pairs preferentially indifferent. Then

$$[I_1^1, I_2^1] \sim [I_1^2, y_2] \tag{8}$$

$$\Leftrightarrow w_1 d_1(I_1^1) + w_2 d_2(I_2^1) = w_1 d_1(I_1^2) + w_2 d_2(y_2) \tag{9}$$

$$\Leftrightarrow \frac{w_1}{w_2} = \frac{d_2(y_2) - d_2(I_2^1)}{d_1(I_1^1) - d_1(I_1^2)} \tag{10}$$

where ~ stands for 'is as good as', i.e. preferentially equal.

Fig. 2 shows the determination of the ratio of the weights. The final weights can be computed when n-1 pairs of attributes have been compared. The sum of weights can be normalised to be 1,

$$\sum_{i=1}^{n} w_i = 1 \tag{11}$$

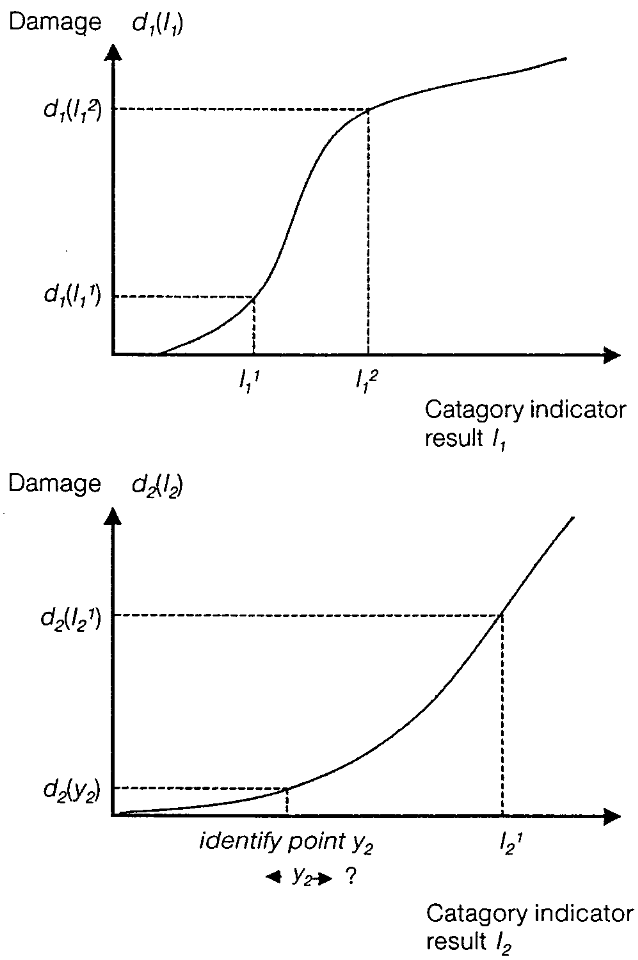

Fig. 2: Elicitation of tradeoff weights

One way of carrying out the tradeoff procedure is to select three extreme points T_1, N_1 and N_2, and to adjust the fourth point y_2 such that indifference is achieved. Then

$$[T_1, N_2] \sim [N_1, y_2] \tag{12}$$

$$\Leftrightarrow \frac{w_1}{w_2} = \frac{d_2(y_2) - d_2(N_2)}{d_1(T_1) - d_1(N_1)} \tag{13}$$

The range of achievement levels for I_1 and I_2 can be denoted as $[T_1, N_1]$ and $[T_2, N_2]$. In addition, the single-attribute functions are normalised onto the [0,1] interval, i.e. the values $d_1(T_1)=0$, $d_2(T_2)=0$, $d_1(N_1)=1$ and $d_2(N_2)=1$ are here assigned to be the best and worst possible consequences.

Assuming that y_2 is determined to be T_2, the ratio of impact category weights can now be expressed by

$$\frac{w_1}{w_2} = \frac{d_2(T_2) - d_2(N_2)}{d_1(T_1) - d_1(N_1)} = \frac{0-1}{0-1} = 1 \tag{14}$$

This leads to the situation in which the weights have equal importance.

The total environmental impact result of the product system a can be calculated according to

$$I(a) = \sum_{i=1}^{n} w_i \cdot \bar{d}_i(I_i(a)) \tag{15}$$

where

$$\bar{d}_i(I_i(a)) = d_i(N_i) - d_i(N_i - I_i(a)) \tag{16}$$

The transformation $\bar{d}_i(.)$ is needed because the contribution of the product system to the damage cannot directly be read from the non-linear damage function (see Fig. 1). This approach allows the shape of the damage function to be taken into account.

It is assumed that the damage (=single-attribute value) functions are linear and pass through the origin. In addition, the damage functions are normalised onto the [0,1] range so that the values $d_1(0)=0$, $d_2(0)=0$, $d_1(N_1)=1$ and $d_2(N_2)=1$ are assigned to the best and worst possible consequences. Given these conditions, the damage functions will satisfy the form (see Seppälä 1997, 1999)

$$\bar{d}_i(I_i(a)) = d_i(I_i(a)) = \frac{I_i(a)}{N_i} \tag{17}$$

Furthermore, the total environmental impact result of product system a is then obtained from

$$I(a) = \sum_{i=1}^{n} w_i \frac{I_i(a)}{N_i} \tag{18}$$

in which the impact category weights reflect the range of damage $d_i(N_i) - d_i(0)$. This means that, in the tradeoff procedure, the aim is to select a point y_i from the attribute I so that $[0, N_i]$ is as good as $[y_i, N_i]$ $(N_i, N_i \in I_i, i = 1,...,n)$.

Assume that the damage functions are linear and they pass I_i axes at indicator result points I_i^{TH}. This means that the damage functions have thresholds. A threshold means that there is zero damage below a category indicator result that is greater than zero (Fig. 3). If the points I_i^{TH} are considered the best consequences, we assign $d_i(I_i^{TH})=0$. In addition, $d_1(N_1)=1$ and $d_2(N_2)=1$ are assigned to worst consequences. The total environmental impact result can now be calculated by

$$I(a) = \sum_{i=1}^{n} w_i \cdot \bar{d}_i(I_i(a))$$

$$= \sum_{i=1}^{n} w_i (d_i(N_i) - d_i(N_i - I_i(a)))$$

$$= \sum_{i=1}^{n} w_i \Big(\big(\frac{1}{N_i - I_i^{TH}} \cdot N_i - \frac{I_i^{TH}}{N_i - I_i^{TH}}\big) - \big(\frac{1}{N_i - I_i^{TH}} \cdot (N_i - I_i(a)) - \frac{I_i^{TH}}{N_i - I_i^{TH}}\big) \Big)$$

$$= \sum_{i=1}^{n} w_i \frac{I_i(a)}{N_i - I_i^{TH}} \qquad (19)$$

in which the impact category weights reflect the range of damage $d_i(N_i) - d_i(I_i^{TH})$. The range corresponds to the range of damage $d_i(N_i) - d_i(0)$ used in the case of Eq. 18. Thus, the tradeoff procedure produces the same attribute weights w_i as in the cases of Eqs. 18 and 19.

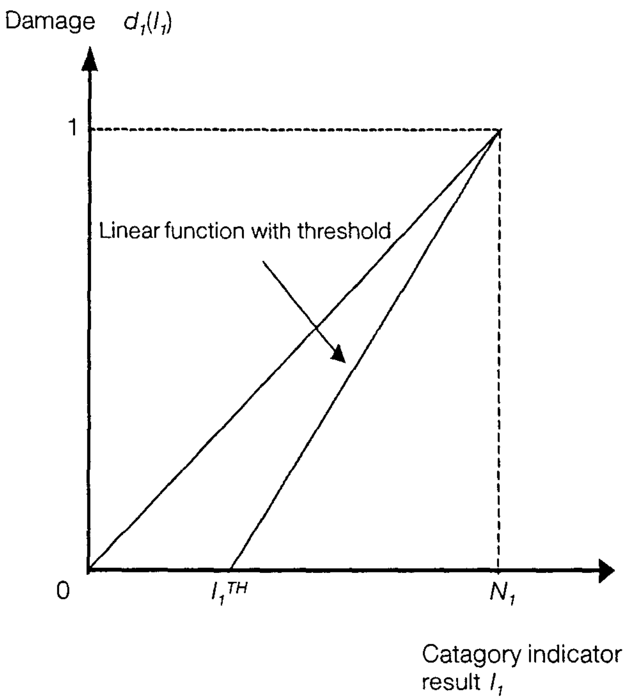

Fig. 3: Linear damage function with threshold

As can be seen in Eqs. 18 and 19, the normalisation factors depend on the shape of the damage function and the range of category indicator results (=attributes) under consideration. In the case of Eq. 18 the normalisation factor is N_i while in the case of Eq. 19 it is $N_i - I_i^{TH}$. Note that differences between the results obtained using Eqs. 18 and 19 are not great if the values of I_i^{TH} are small compared to the values of N_i.

3 Comparison of methods

3.1 Ecoindicator 95

In the Ecoindicator 95 method it is assumed that damage functions are linear and pass through the origin (→ **Fig. 4**).

The definition of targets T_i is such that the non-zero targets with equal importance hold, i.e. $d_1(T_1)$ is as good as $d_2(T_2)$. The targets can be obtained from expert judgements based on scientific data (see Goedkoop 1995).

Choosing the working point to be T_i in the tradeoff procedure, the impact category weights can be calculated

$$\frac{w_2}{w_1} = \frac{d_1(y_2) - d_1(T_1)}{d_2(0) - d_2(T_2)} \qquad (20)$$

where the category indicator result y_2 is determined according to the preference indifference

$[0, N_2] \sim [N_1, y_2]$.

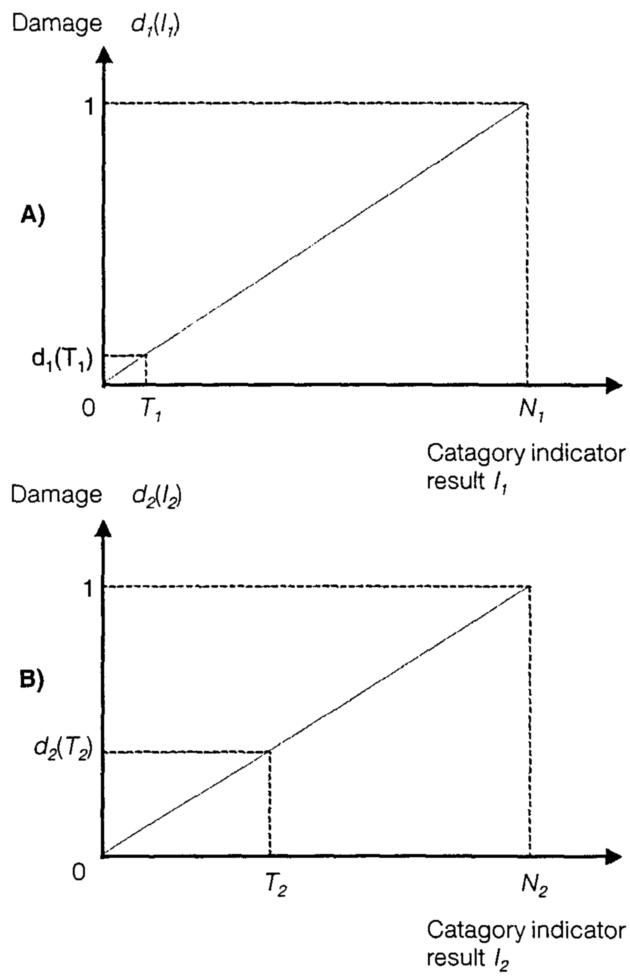

Fig. 4: Two different reduction factors in the Ecoindicator 95 method described in the damage function framework. In Figs. 4A and 4B the damages caused by targets T_1 and T_2 have equal preferences, i.e. $d_1(T_1)$ is as good as $d_2(T_2)$

Point y_2 is chosen to be 0 because, in the Ecoindicator 95 method, it is assumed that $d_1(T_1) \sim d_2(T_2)$. Thus, equal weights hold. According to the MAVT framework, the total environmental impact result can be written as

$$I(a) = \sum_{i=1}^{n} \frac{I_i(a)}{T_i} = \sum_{i=1}^{n} \frac{N_i}{T_i} \cdot \frac{I_i(a)}{N_i} \qquad (21)$$

which corresponds to the calculation rule of the Ecoindicator 95 method. Note that the last part of Eq. 21 shows that weights must be interpreted as reduction factors when normalisation factors N_i are used.

However, the scientific bases for using linear damage functions passing through the origin are rather weak. The use of critical loads in environmental science (e.g. critical loads for acidification and eutrophication (Posch et al. 1999)) means that the corresponding damage functions have thresholds. However, it is difficult to define real values for thresholds and the correct shapes for damage functions. If we accept the thresholds for the linear damage functions as presented in Fig. 3, the MAVT framework leads to (see Eqs. 18 and 19)

$$I(a)=\sum_{i=1}^{n}\frac{N_i}{T_i}\cdot\frac{I_i(a)}{N_i-I_i^{TH}} \tag{22}$$

where there is zero damage below category indicator result I_i^{TH}. Eq. 22 differs from Eq. 21 but differences between the results obtained using the two equations are not great if the values of I_i^{TH} are small compared to the values of N_i.

3.2 Environmental theme method

The structures of the Environmental Theme (ET) method and the Ecoindicator 95 method are similar. Equation 21 derived from MAVT corresponds to Eq. 3, in which the distance-to-target weights of the Environmental Theme (ET) method (Eq. 4) are used. This means that the ET method is based on the use of linear damage functions passing through the origin. Furthermore, this means that the ET method is consistent with the MAVT framework if we have equal preferences for damages caused by non-zero target values T_i, i.e. $d_1(T_1)\sim d_2(T_2)\sim,\ldots,\sim d_n(T_n)$.

In practice, targets T_i are defined either as ecological critical loads or as politically maximum acceptable limits in the ET method (see Baumann and Rydberg 1994). Targets T_i are equally important when the targets are determined according to ecological critical loads. However, the use of critical loads means that the corresponding damage functions have thresholds, i.e. $T_i=I_i^{TH}$. According to MAVT, this leads to the calculation rule

$$I(a)=\sum_{i=1}^{n}\frac{N_i}{T_i}\cdot\frac{I_i(a)}{N_i-T_i} \tag{23}$$

which differs from Eq. 21.

In the case of political targets, we do not normally have equal preferences for the damages caused by target values T_i from the point of view of environmental impacts. The different preferences lead to the situation in which

$$\frac{w_l}{w_t}\neq\frac{\dfrac{N_l}{T_l}}{\dfrac{N_t}{T_t}}\ ,\ l,t\in i \tag{24}$$

Thus, the calculation rule is not consistent with Eq. 21 and the MAVT framework.

3.3 Ecoscarcity method

In the Ecoscarcity method the normalisation factor can be interpreted as target levels of environmental interventions j related to a given region (L_j^T). Furthermore, the intervention weights are the reduction factors (L_j^N/L_j^T). Assuming the simple conditions (linear damage functions passing through the origin and equal damages $d_i(L_j^T)$ hold at target points L_j^T), the MAVT framework produces the following calculation rule for the total environmental impact result (see Eq. 21):

$$I(a)=\sum_{j=1}^{m}\frac{L_j(a)}{L_j^T}=\sum_{j=1}^{m}\frac{L_j^N}{L_j^T}\cdot\frac{L_j(a)}{L_j^N} \tag{25}$$

In this case intervention weights (w_i^L) are equal and the equation does not correspond to the aggregation rule of the Ecoscarcity method (Eq. 1). The reduction factors (L_j^N/L_j^T) of the Ecoscarcity method need the use of normalisation factors L_j^N from the point of view of MAVT. Note that the case of Eq. 25 is analogous to that of Eq. 21.

Assume that if the targets L_j^T are determined according to ecological critical loads. The MAVT framework produces the following solution (see Eq. 23):

$$I(a)=\sum_{j=1}^{m}\frac{L_j^N}{L_j^T}\cdot\frac{L_j(a)}{L_j^N-L_j^T} \tag{26}$$

This does not correspond to the calculation rule of the Ecoscarcity method.

4 Discussion and Conclusions

The specific framework of the interpretation of distance-to-target methods that is addressed in this paper is the multiattribute value theory (MAVT) of decision analysis. There are also other multicritèria approaches, e.g. the analytical hierarchy process AHP (Saaty 1990) and Electre-type methods (Roy 1990). However, Electre-type methods and the AHP have not been able to offer a widely accepted axiomatic foundation for value measurement of weights so far and therefore will not be discussed here. Note that Salo and Hämäläinen (1997) have shown that the questioning in the AHP method can be modified so that the results will be similar to those of MAVT.

On the basis of MAVT we can answer questions such as

- What aggregation rule is theoretically required for the calculation of a total environmental impact result?
- What does the concept of a weight mean and how should the weights be assessed?

Therefore, considering the distance-to-target weighting methods in the MAVT setting provides an alternative viewpoint into the debate about weighting and normalisation in life cycle impact assessment. Furthermore, the simple additive

aggregation model based on MAVT can be used in life cycle impact assessment (LCIA).

In MAVT, impact category weights are interpreted as scaling constants, which reflect the tradeoffs between the value units of different category indicator results. How many times more important is it to decrease domestic emissions causing eutrophication than acidification? The answer clearly depends on the relevant ranges of category indicator results and the current levels of the impacts.

This paper describes weight evaluation in life cycle impact assessment by the tradeoff procedure. It is only one method for weight elicitation. There are many other procedures for the determination of weights in the MAVT literature, e.g. the ratio method (Edwards 1977), the swing weighting method (von Winterfeldt and Edwards 1986) and the pricing out method (Keeney and Raiffa 1976). In practice, the tradeoff method is difficult and time consuming to use compared with the other methods (Borcherding et al. 1991). However, its advantage is the strong theoretical foundation (Keeney and Raiffa 1976, Weber and Borcherding 1993). This is why we have therefore illustrated weighting in the context of the tradeoff procedure. The aim of this paper was not to evaluate the features of different elicitation techniques in the LCA context. There is a continuing need for further research into applying different elicitation techniques in LCA.

In LCIA it is often assumed that a product system *a* under investigation causes a small load compared with a background load caused by all other society's activities in a given area. By taking into account the product system specific ranges in the category indicator results the tradeoff procedure means that values of impact category weights reflect better the case of product system *a*. However, it can be assumed that subjects performing the evaluation are not able to make sufficient adjustments when the ranges are too small to understand in the context of impact categories. Furthermore, large ranges allow case-by-case impact category weighting sets to be generated directly from the generic weighting sets. Therefore, there is no need for a separate impact category weighting task in specific LCA studies.

MAVT can also offer a clarification of whether or not to use normalisations in LCIA. Following MAVT's rules, we can obtain different normalisation factors depending on the shape of the damage function and the working point under consideration. As can be seen in this article, the aim of normalisation is to convert the different scales of category indicator results into the same [0,1] range, which is an essential stage before weighting.

In this work, damage functions in the context of the LCIA framework proposed by SETAC and ISO were determined to describe relationships between category indicator results and the effects (damages). An alternative approach is to use damage function mapping interventions into the effect scales. Seppälä (1997, 1999) showed that this leads to the same outcome when calculating the total impact in the way presented here under the assumed simple conditions. However, the alternative approach is likely to be better in capturing the non-linear effects.

Although the scientific bases for using linear damage functions are rather weak, the use of linear damage functions leads to a convenient way of calculating the total impact from the various impact categories into a single score. The MAVT framework presented in this paper also offers a general solution to the weighting as well as to the normalisation and aggregation in the case of non-linear damage functions.

In summary, the study shows that under certain conditions two of the three commonly used impact assessment methods (Ecoindicator 95, ET-method) applying the distance-to-target weighting are consistent with the impact assessment framework derived from MAVT. This consistency holds if non-zero targets with equal importance and linear damage functions passing through the origin are assumed. However, the scientific bases for using linear damage functions without thresholds are weak.

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Relationships Between Weighting Factors and Normalisation in Life Cycle Impact Assessment

Jyri Seppälä[1] and Raimo P. Hämäläinen[2]

[1] Finnish Environment Institute, PB 140, 00251 Helsinki, Finland
[2] Systems Analysis Laboratory, Helsinki University of Technology, Otakaari 1 M, 02150 Espoo, Finland

In Life Cycle Assessment, practitioners sometimes wish to use weighting factors of impact categories in order to obtain a single comparison metric for results of Life Cycle Impact Assessment (LCIA). Many methods, such as economic, panel and distance-to-target approaches, can be used to obtain weighting factors. Different methods produce different weighting factors depending on several factors. One important issue is how the weighting factors are understood. For example, in the panel method we can use an elicitation technique called ratio estimation (see von Winterfeldt and Edwards 1986) with the question form "How much do you prefer to restrict emissions causing acidification as compared to those causing aquatic eutrophication (or vica versa)?" However, the question form does not constitute an acceptable standpoint for a procedure of preference elicitation if emissions are not specified. Are we talking about emissions from Finland or from the whole of Europe?

Assume that we have weighting factors obtained form the weighting task in which different impact category effects caused by Finnish emissions were compared to each other. According to multiattribute value theory (MAVT) the use of these weighting factors in the aggregation rule commonly used in LCIA requires normalisation, in which reference values are calculated on the basis of the Finnish emissions (Seppälä 1997, 1999). Thus,

$$I(a) = \sum_{i=1}^{n} w_i^F \cdot \frac{I_i(a)}{N_i(F)} \tag{1}$$

where $I(a)$ is a total environmental impact caused by product system a, w_i^F is a weighting factor (or weight) of impact category i ($i = 1,\dots,n$) related to adverse effects caused by emissions from Finland, $I_i(a)$ is an indicator result of impact category i caused by product system a and $N_i(F)$ is a reference value (or normalisation factor) of impact category i caused by emissions from Finland.

If impact category effects caused by European emissions are weighted against each other in the weighting task, we get weighting factors w_i^E related to European emissions instead of w_i^F. According to MAVT the use of these weighting factors in the aggregation needs reference values $N_i(E)$ calculated by emissions from Europe.

Equation 1 offers an explanation for the calculation rule of total environmental impact $I(a)$ which does not seem to include normalisation, i.e.

$$I(a) = \sum_{i=1}^{n} w_i \cdot I_i(a) \tag{2}$$

This calculation rule is consistent with MAVT if the weighting factor w_i consists of weighting and normalisation factors. For example, in the case of Equation 1 this means that $w_i = w_i^F / N_i(F)$.

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